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# Modelling of point and non-point source pollution of nitrate with SWAT in the river Dill, Germany

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**Abstract.** We used the Soil and Water Assessment Tool (SWAT) to simulate point and non-point source pollution of nitrate in a mesoscale mountainous catchment. The results show that the model efficiency for daily discharge is 0.81 for the calibration period (November 1990 to December 1993) and 0.56 for the validation period (April 2000 to January 2003). The model efficiency for monthly nitrate load is 0.66 and 0.77 for the calibration period (April 2000 to March 2002) and validation period (April 2002 to January 2003), respectively. However, the model efficiency for daily loads is low (0.15), which cannot only be attributed to the quality of input data of point source effluents. An analysis of the internal fluxes and cycles of nitrogen pointed out considerable weaknesses in the models conceptualisation of the nitrogen modules which will be improved in future research.

## 1 Introduction

Eco-hydrological simulation models like the Soil and Water Assessment Tool (SWAT) (Arnold et al., 1998) are essential tools for decision support in water resources planning (e.g. to meet the requirements of the EU Water Framework Directive). We use SWAT within the framework of the collaborative research center (CRC 299) “Land use options for peripheral regions” at the University of Giessen to assess the impact of potential land use options on both the water quantity and quality.

Prior to the application of SWAT to predict the influence of land use change on river water quality, the model is tested to observed data of discharge as well as nitrate load of the river Dill. Both the accuracy and the reliability of the SWAT model with respect to nitrogen cycling are discussed in detail in this paper.

## 2 Materials and methods

### 2.1 The Dill catchment

The Dill catchment is located in mid-Hesse, Germany, and is part of the “Lahn-Dill-Bergland” (see Fig. 1). The geology of the catchment consists of carboniferous clay shists and graywacke in the south, basic devonian volcanic rocks in the center and quarzitic devonian sandstones and clay shists in the north. The dominant landform process during the pleistocene was solifluction, thus, the rocks are widely covered by periglacial deposits. Holocene colluviums appear mainly in river valleys. The dominant soil type is a shallow cambisol (Sauer, 2002).

The Dill catchment faces a decline of farming, which causes an increase of fallow land. Approximately 9% of the catchment is fallow land, based on remotely sensed data in 1994 (Nöhles, 2000). Arable land contributes to about 7% in the catchment. Table 1 summarises some of the main characteristics of the study site.

### 2.2 The SWAT model

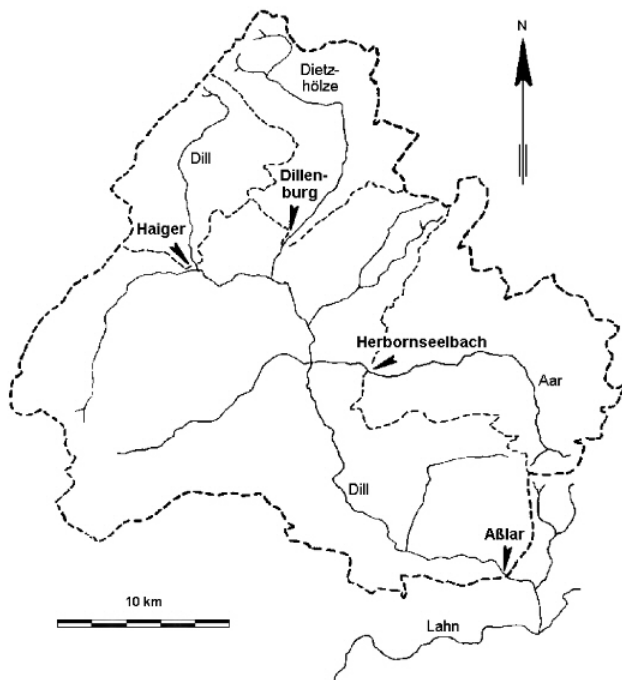
SWAT is a semi-distributed, process oriented hydrological model. It is a continuous time model which simulates both the water balance and the nutrient cycle with a daily time step. The preprocessing of the spatially distributed data is achieved in a two-step approach. First the subbasins, stream network, channel length and hill slopes are derived from a digital elevation model (DEM). Second, homogeneous land use and soil classes are overlaid to yield multiple hydrological response units (HRU) within each subbasin. Interpolation of weather data is achieved through the nearest-neighbour method, i.e. each subbasin receives precipitation from one single rain gauge nearest to its centroid. Rainfall within one subbasin is corrected for height with 10 elevation bands.

SWAT simulates evapotranspiration, infiltration, percolation, runoff generation, nutrient cycling and transport for each HRU. Water and sediment routing as well as in-stream

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**Table 1.** Characteristics of the Dill catchment. Information on topography is derived from the DEM, land use distribution is taken from processed LANDSAT TM5 scenes (Nöhles, 2000) and climate data are taken from the hydrologic yearbook.

Parameter	Value	Parameter	Value
Topography		Land use	
Area	692 km <sup>2</sup>	Agriculture	7%
Maximum elevation	675 m a.s.l.	Pasture	21%
Minimum elevation	155 m a.s.l.	Changing vegetation	9%
Mean slope	8°	Forest, evergreen	30%
Maximum slope	35.2°	Forest, deciduous	25%
Climate		Urban area	8%
Annual precipitation	960 mm	Water	<1%
Annual discharge	457 mm	Demography	
Annual temperature	8.3°C	Population	148 000



**Fig. 1.** Stream network and locations of gauged subbasins within the Dill catchment.

nutrient processes are simulated along the channel length for each subbasin (Neitsch et al., 2002). The algorithms for nitrogen cycling and transport are based on the EPIC model (Erosion-Productivity Impact Calculator) (Williams et al., 1984). Net mineralisation is simulated with one active and one stable organic nitrogen pool. Plant uptake of nitrogen is estimated using a supply and demand approach. Nitrate in the soil can be removed from the soil via denitrification, mass flow of water as well as plant uptake (Neitsch et al., 2002). In this study we used the SWAT-G model (Eckhardt et al., 2002), an adaption of SWAT2000, which considers the anisotropy of vertical and lateral hydraulic conductivity in mountainous regions.

### 2.3 Input data

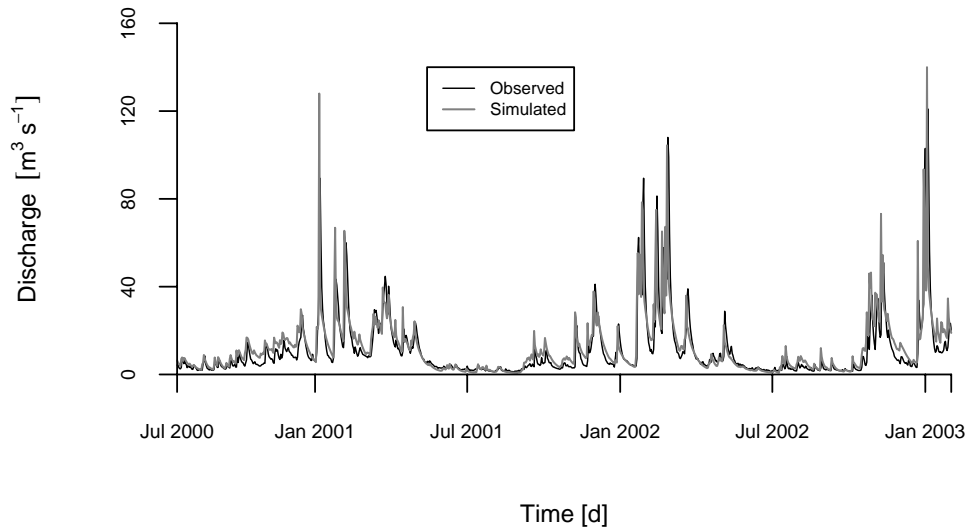
The topography of the catchment was derived from a 25×25 m DEM. The land use map was compiled from LANDSAT TM5 images taken in 1994 (Nöhles, 2000) and the soil map is based on digitised soil maps with a scale of 1:50 000 (Hessisches Landesamt für Umwelt und Geologie, 2000).

The meteorological input data were obtained from the German Weather Service (DWD). The precipitation data consisted of homogenised daily records of 12 precipitation gauges. Furthermore, daily records of solar radiation, minimum and maximum temperature, humidity as well as wind speed were taken from 2 climate stations. Data of monthly point source effluents were taken from 3 municipal sewage treatment plants and one steel mill, where nitric acid is used to harden steel. These four point sources comprise about 90% of total nitrate effluents from point sources in the catchment (Lenhart, 2003).

A three years crop rotation was simulated on arable land, where winter rape, winter barley and oats were fertilised with 200, 120 and 80 kg N ha<sup>-1</sup>, respectively. Pasture was fertilised in the beginning of March and June as well as in mid-July and mid-November with a total of 160 kg N ha<sup>-1</sup> (Lenhart, 2003). A constant nitrate concentration in precipitation of 1.5 mg N l<sup>-1</sup> was assumed.

### 2.4 Discharge and nitrate load measurements

An automatic sampler (ISCO 3700) was installed in April 2000 at the river gauge Asslar (R 3462100, H 5605350). Daily composite samples of hourly subsamples are taken and analysed in the laboratory. Lenhart (2003) used a photometric method to analyse NO<sub>3</sub>-N for the period April 2000 until April 2002. From May 2002 onwards, the samples are analysed with an ionchromatograph (DIN EN ISO 10304-1, 1995). A simple linear regression was used to account for the systematic difference between the two methods and to



**Fig. 2.** Simulated and observed daily discharge [ $\text{m}^3 \text{s}^{-1}$ ] for the validation period (April 2000 to January 2003) at the gauge Asslar.  $E=0.56$ ,  $b=0.81$ ,  $R^2=0.60$ .

homogenise the series of nitrate concentrations,

$$\text{NO}_3 \text{ IC} = \text{NO}_3 \text{ PH} (0.87 \pm (0.07)) \quad R^2=0.72 \quad (1)$$

where  $\text{NO}_3 \text{ IC}$  and  $\text{NO}_3 \text{ PH}$  are  $\text{NO}_3\text{--N}$  concentrations measured with the ionchromatograph and the photometer, respectively. Daily nitrate load is obtained by multiplying daily average discharge with daily average nitrate concentration.

### 2.5 Calibration

Since nitrate fluxes strongly depend on water fluxes, parameters controlling water balance were calibrated in a first step. We used the Shuffled Complex Evolution algorithm developed at the University of Arizona (SCE-UA) for automatic calibration of discharge in the period from November 1990 to December 1993. The SCE-UA algorithm has proven to be a powerful tool for finding the global minimum in the parameter space defined by the user (Duan et al., 1992). The optimisation procedure as well as the chosen parameters are identical to those reported by Huisman et al. (2004).

In a second step, the period from April 2000 to March 2002 was used to manually calibrate daily nitrate load. The initial and final values of the calibrated parameters are given in Table 2. These parameters were chosen based on the sensitivity analysis of Lenhart et al. (2002).

## 3 Results and discussion

Figure 2 shows simulated and observed daily discharge at the river gauge Asslar for the validation period. The model efficiency ( $E$ ) after Nash and Sutcliffe (1970) for daily discharge is 0.81 and 0.56 for the calibration and validation period, respectively. In general the model fits the observed rise of discharge as well as the recession well. However, the timing and the height of single flood peaks in January 2001 and January

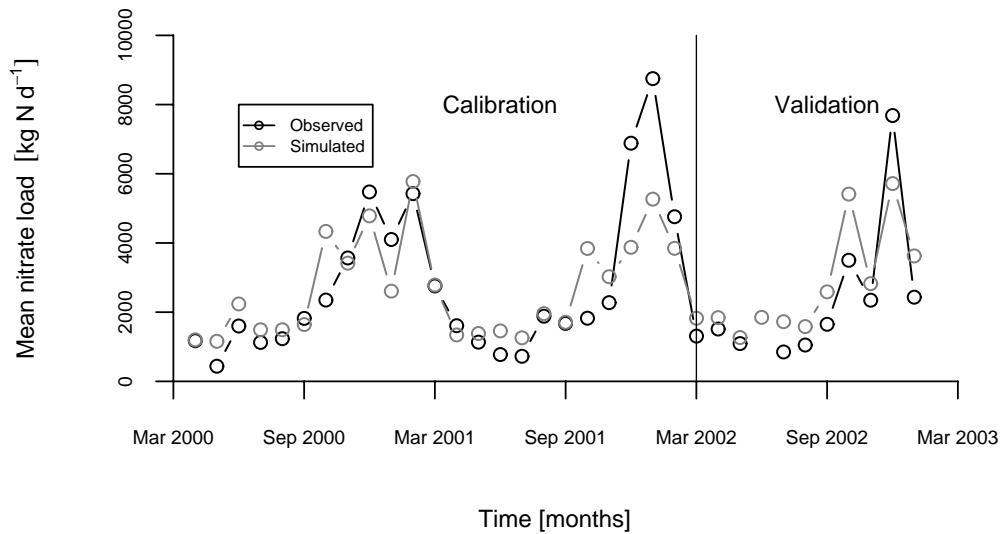
**Table 2.** Manually calibrated parameters which control both nitrogen cycle and transport.

Parameter	Process	Initial	Final
CMN	Humus mineralisation	0.01	0.003
RSDCO	Residue mineralisation	0.10	0.005
NPERCO	Nitrate transport	0.2	0.4

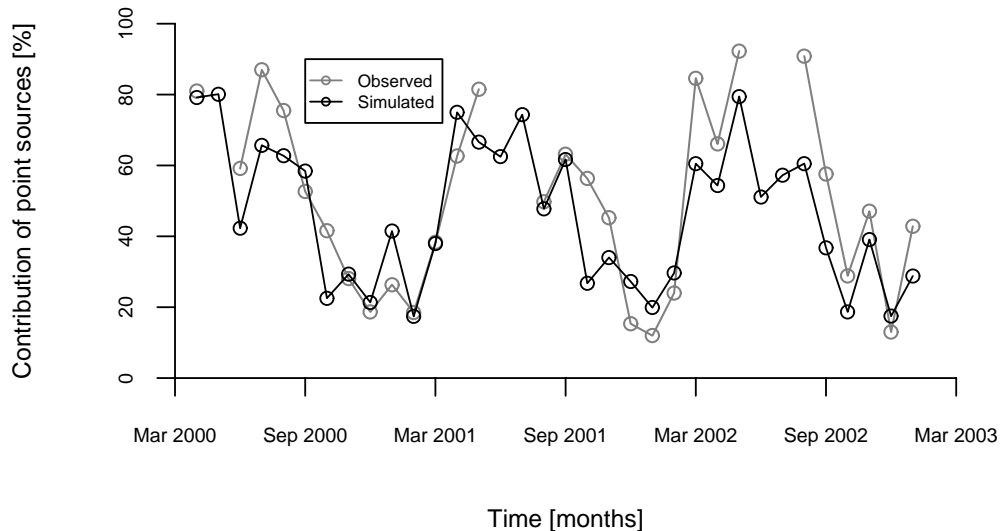
2003 are less accurately predicted. This can be attributed to the model structure of SWAT. Since it is a continuous time model with a daily time step, subscale processes such as flood generation can not be efficiently simulated. Furthermore, there is a temporal mismatch between daily precipitation measured for 24 h beginning from 7:30 AM and daily discharge averaged over 24 h from midnight on.

Figure 3 gives simulated and observed monthly nitrate loads for both the calibration and validation period. Table 2 shows the coefficient for humus mineralisation (CMN) and the coefficient for residue mineralisation (RSDCO), which needed to be reduced by one to two orders of magnitude from the default values to slow down the simulated kinetics. For the calibration and validation period  $E$  is 0.66 and 0.77, respectively. The peak load of nitrate occurs during winter-time. This can be attributed to both the high runoff during this season, which leads to increased nitrate transport as well as the lack of plant uptake, which favours the generation of leachable nitrate. SWAT simulates the seasonal cycle accurately. However, SWAT did not match the temporal patterns of daily loads, thus the model efficiency for daily load is low ( $E=0.15$ ).

The contribution of point source pollution during low flow phases in summer can account for up to 90% of the total nitrate load at the outlet. In winter the contribution is lower



**Fig. 3.** Simulated and observed monthly nitrate load [ $\text{kg N d}^{-1}$ ] for the period April 2000 to January 2003 at the gauge Asslar. Calibration:  $E=0.66$ ,  $b=1.26$ ,  $R^2=0.69$ . Validation:  $E=0.70$ ,  $b=1.21$ ,  $R^2=0.77$ .

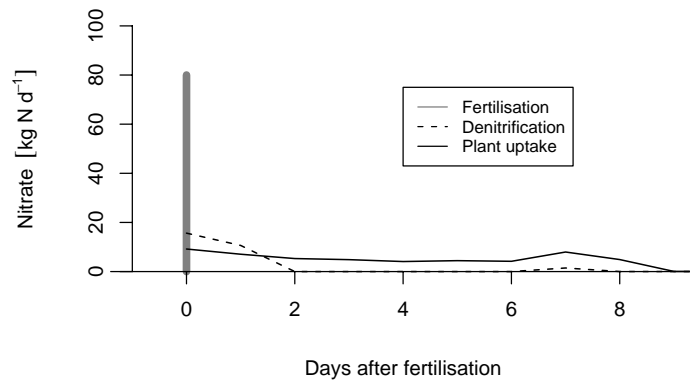


**Fig. 4.** Monthly contribution of point sources as a fraction of observed and simulated total nitrate load at the gauge Asslar.

(Fig. 4). We assumed a simple mixing model to derive the ratio between the sum of point source inlets along the channel and the total load at the river outlet. It should be mentioned, that during summertime the ratio between total effluents and observed nitrate load can exceed 100% which is indicated in Fig. 4 by missing data. This can be attributed to the simple mixing model, which ignores degradation processes in the river, as well as the considerable uncertainty of the aggregated point source inlet data.

The model performance regarding nitrate load prediction is in accordance with reports from other authors using SWAT on various catchments. Lenhart et al. (2003) worked on the same catchment and obtained a model efficiency for the validation period (April 2001 to March 2002) of 0.09 and 0.31 for daily and monthly loads, respectively. In contrast

to the present study, where point source effluents were used as input data for the simulation, Lenhart et al. (2003) used annual estimated point source effluents based on a correlation with population data. These loads were used to calculate a point source background concentration by division with simulated daily runoff. This background concentration was subtracted from the observed concentrations. The adjusted concentration was then used to calculate daily loads for validation and calibration. Chaplot et al. (2004) predicted mean monthly nitrate loads in the Walnut Creek watershed ( $51.3 \text{ km}^2$ ), Iowa. The coefficient of determination ( $R^2$ ) in their work is 0.73. Santhi et al. (2001) found a similar agreement between simulated and observed monthly data ( $R^2=0.72$ ,  $E=0.64$ ) when they used SWAT to predict nitrate loads in the Bosque River watershed ( $4277 \text{ km}^2$ ), Texas.



**Fig. 5.** Denitrification and plant nitrate uptake of winter barley after fertilisation with  $80 \text{ kg N ha}^{-1}$ .

However, Grizzetti et al. (2003) found a lower agreement ( $E=0.30$ ) when they used SWAT at the Vantaanjoki watershed ( $1680 \text{ km}^2$ ), Finland, to model diffuse emissions and retentions of nutrients on a monthly basis.

As outlined above, we achieved a similar simulation efficiency of nitrate load at the catchment outlet as other authors working with SWAT. However, we examined the internal fluxes and cycles of nitrate in the catchment in addition. In Fig. 5 the rates of denitrification and plant uptake are given for one HRU where winter barley grows on agricultural land. Within the first two days after fertilisation of  $80 \text{ kg N ha}^{-1}$  about  $26 \text{ kg N ha}^{-1}$  are denitrified and about  $50 \text{ kg N ha}^{-1}$  nitrate are taken up by the plant in the first nine days. Both the amount and kinetics are surprisingly high. The high kinetics in plant uptake can be explained by the lack of a sink limitation for the simulated nitrate uptake. If the nitrate content of the soil does not supply the nitrate demand of the crop, a deficiency will occur and accumulate as long as this condition continues. As soon as the nitrate pool in the soil is increased by fertilisation, the plant can take up as much as its accumulated deficiency, causing the high kinetics.

Table 3 gives average annual values for the main nitrate fluxes in the same HRU as used for Fig. 5. It is obvious that the amount of simulated denitrification is far too high. In SWAT denitrification occurs when soil moisture exceeds 95% of field capacity. Since the soils are not highly conductive and the water is often perched on the underlying bedrock layer, this threshold is often exceeded. Hence, leaching and denitrification are two competing processes within the model.

## 4 Conclusions

We used SWAT to simulate both point and non-point source nitrate pollution in a mesoscale catchment. The model was able to accurately simulate the seasonal cycle of nitrate load. The agreement between simulated and observed daily nitrate load was lower. The poor accuracy of simulated daily loads can not only be explained by the limited availability of daily

**Table 3.** Average annual values for the main nitrate fluxes on an example HRU with agricultural land use.

Process	Nitrate $\text{kg N ha}^{-1}$
Fertilisation	151.7
Denitrification	135.5
Plant uptake	44.3
Lateral flow	23.9
Leaching	4.4
Surface runoff	5.1

data of point source effluents. Since denitrification rates and daily plant uptake are misspredicted, the conceptualisation of the internal nitrogen cycling seems to be the main limitation. Clearly, this model set up can not yet be used for scenario analysis. Besides an accurate result for the outlet, a consistent representation of internal N cycling is also required. Otherwise, scenario analysis might result in unrealistic predictions of changes in N budgets due to land use change.

In our ongoing research, the algorithms of organic litter distribution between 3 different pools, gross mineralisation, ammonification, denitrification as well as plant uptake considering a source and sink limitation from the Denitrification-Decomposition Model (DNDC) (Li et al., 1992, 2000; Zhang et al., 2002) are implemented and tested in SWAT to overcome the aforementioned problems.

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